Report

Fall-run Chinook Salmon spawning assessment during 2013 and 2014 within the San Joaquin River, California

Annual Technical Report

Fall-run Chinook Salmon spawning assessment during 2013 and 2014 within the San Joaquin River, California

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EXECUTIVE SUMMARY

After the construction of Friant Dam in the 1940's, fall- and spring-run Chinook Salmon (*Oncorhynchus tshawytscha*) were extirpated from the lower San Joaquin River upstream of the confluence with the Merced River. Currently, the San Joaquin River Restoration Program (SJRRP) is working towards restoring the river, and maintaining naturally-reproducing and self-sustaining populations of Chinook Salmon. Although there is consensus among managers that the lack of river connectivity (i.e., flow) is a restoration priority, there is uncertainty regarding the existing quality and quantity of suitable salmon spawning habitat and the value of trap-and-haul techniques within the Restoration Area to meet SJRRP restoration goals. Here, we evaluated the spawning success of adult fall-run Chinook Salmon translocated from upstream of Hills Ferry Barrier and associated sloughs to Reach 1 of the Restoration Area upstream of the Highway 99 bridge. We conducted a redd and carcass visual observation survey in Reach 1 to assess the spawning success of translocated salmon during the 2013 (October 2013 to March 2014) and 2014 (October 2014 to March 2015) field seasons. Using emergence traps, we also assessed the number of fry emerging from a total of fifteen redds detected during the extent of the study. Our results revealed that the spawning success of translocated females was 63% in 2013 and 67% in 2014. We estimated that a total of 189 (150–282; 95% confidence limits) adult translocated salmon were available for spawning in 2014. Assuming that 44% of the adults were females (observed among recovered carcasses), we projected that a total of 87 translocated females had the potential to spawn (i.e., escape), which is consistent with the number of redds (n= 81) detected in 2014. We observed, on average, 1,277 salmon fry emerge from redds monitored in 2013 and 997 salmon fry emerge from redds monitored in 2014. The Fisheries Management Plan (SJRRP 2010) set restoration targets of no more than 15% pre-spawn mortality and minimum 50% egg to fry survival based on a fecundity of 4200 eggs per female. Our study revealed that the spawning success of translocated Chinook Salmon in Reach 1 of the Restoration Area during 2013 and 2014 was below the SJRRP 85% target and that the number of fry produced from their redds was below the SJRRP target of 2,100 fry per redd. As a result, this study provides evidence that the quality of extant salmon spawning habitat within the Restoration Area or the effectiveness of adult translocation activities may be inadequate for achieving the Chinook Salmon reproduction targets at least during critical dry water years. Although further research is needed, the information gathered during this study should be used to help inform fall-run and spring-run reintroduction activities, inform flow management decisions concerning habitat suitability, guide potential habitat restoration, and evaluate the effectiveness of ongoing trap and haul techniques.

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Abbreviations and Acronyms

CJS	Cormack-Jolly-Seber
SJRRP	San Joaquin River Restoration Program
SIG	Small Interdisciplinary Group
USBR	United States Bureau of Reclamation
CDFW	California Department of Fish and Wildlife
USEPA	United States Environmental Protection Agency
HFB	Hills Ferry Barrier
ATU	accumulated thermal unit
FL	fork length
rkm	river kilometer
CDEC	California Data Exchange Center

1.0 Fall-run Chinook Salmon spawning assessment during 2013 and 2014 within the San Joaquin River, California

1.1 Introduction

Historically, the mainstem San Joaquin River supported the southernmost populations of spring-, fall-, and late-fall run Chinook Salmon *Oncorhynchus tshawytscha* (Fry 1961; Fisher 1994; Yoshiyama et al. 2001). Unfortunately, the construction of Friant Dam during the 1940s coupled with excessive water diversions to support agriculture, hydropower, and flood control have led to considerable habitat degradation and fragmentation within the mainstem San Joaquin River (Yoshiyama et al. 2001; Williams 2006). These alterations have been attributed to the extirpation of all salmon runs within the San Joaquin River above the confluence of the Merced River (Yoshiyama et al. 2001). However, fall-run Chinook Salmon still occur in the major tributaries of the lower San Joaquin River including the Merced, Tuolumne, Stanislaus, Mokelumne, and Cosumnes rivers (Yoshiyama et al. 2000).

The San Joaquin River Restoration Program (SJRRP) is working towards restoring flows and river connectivity, and reestablishing naturally-reproducing and self-sustaining populations of fall- and spring-run Chinook Salmon to the San Joaquin River between Friant Dam and the confluence of the Merced River (henceforth Restoration Area; Figure 1; SJRRP 2015). To accomplish this, the SJRRP has established several Chinook Salmon population targets to guide restoration and achieve salmon population viability within the Restoration Area (SJRRP 2010). A critical component to assessing population viability and guiding restoration is knowing the recruitment (i.e., spawning) success of individuals within a population and understanding why it varies, respectively (Williams et al. 2002). The SJRRP's Fisheries Management Plan (SJRRP 2010) states that spawning failure should not exceed 15% (e.g., 85% of females create redds) and that at least an average of 2,100 fry (i.e., 50% of average fecundity) should be produced from each female (i.e., redd) to achieve the SJRRP’s preliminary population targets.

There are several potential factors that may limit the success of spawning and the survivorship of Chinook Salmon from the egg to fry life-stages within Reach 1 of the Restoration Area (SJRRP 2009). These stressors include recreational harvest or poaching, predation, river flow, water temperature, dissolved oxygen, substrate size and composition, and water depth and velocity (Figure 2; SJRRP 2009). Although the initial assessments of spawning and incubation habitat using egg tubes embedded in artificial redds have indicated that the quantity and quality may be inadequate to achieve current Chinook Salmon population targets (SJRRP 2012a), there is considerable uncertainty regarding the success of natural spawning within the Restoration Area given the existing habitat.

Currently, the SJRRP has not completed the fish passage projects or maintained adequate river flows within the Restoration Area necessary to allow volitional salmon passage (SJRRP 2012b, 2015). As a result, salmon are unable to migrate to or from extant spawning habitat within the Restoration Area without management intervention. Thus, the SJRRP has been exploring the feasibility and effectiveness of translocating salmon around existing barriers and unsuitable habitat while restoration projects are in progress. In 2012, the SJRRP demonstrated that transporting adult Chinook Salmon was a feasible management alternative (Jackson 2012), which can be used in the interim to advance our understanding of the system and develop robust management alternatives to ameliorate limiting factors (SJRRP 2010). Although there is a consensus among managers that the lack of river connectivity (i.e., flow) is a restoration priority, there is considerable uncertainty regarding the value of interim fish passage intervention to meet restoration goals (e.g., trap-and-haul; SJRRP 2010) and quality of extant salmon spawning habitat within the Restoration Area (SJRRP 2010; SJRRP 2015).

To prevent the straying of fall-run Chinook Salmon into the Restoration Area from other major tributaries of the San Joaquin River, the California Department of Fish and Wildlife (CDFW) constructs the Hills Ferry Barrier (HFB) annually and maintains it from late September to late December. Salmon that circumvent HFB and enter the Restoration Area are exposed to poor habitat quality and currently do not have access to suitable spawning habitat (Hallock and Van Woert 1959; Reynolds et al. 1993; Portz et al. 2011). Because fish passing this point would otherwise be “lost” in the system, there was an opportunity to translocate these fall-run Chinook Salmon into Reach 1 of the Restoration Area during the fall of 2013 and 2014 to evaluate the effectiveness of existing adult trap-and-haul techniques, develop a monitoring framework for salmon spawning assessments, and improve our understanding of salmon spawning success within the Restoration Area.

1.1.1 Objectives:

The fundamental objective of this report was to describe the overarching success of naturally spawning fall-run Chinook Salmon translocated from Reach 5 to Reach 1 of the Restoration Area during the 2013 (October 2013 to March 2014) and 2014 (October 2014 to March 2015) field seasons. We studied translocated adult fall-run Chinook Salmon collected upstream from HFB with the following objectives:

1. Quantify the spawning success of translocated females, and determine the spatial and temporal distribution of spawning
2. Estimate the number of translocated salmon that were available to spawn
3. Determine the number of fry produced among observed redds
4. Relate fry production within observed redds to environmental variables hypothesized to affect the survival probability from egg to emerging fry

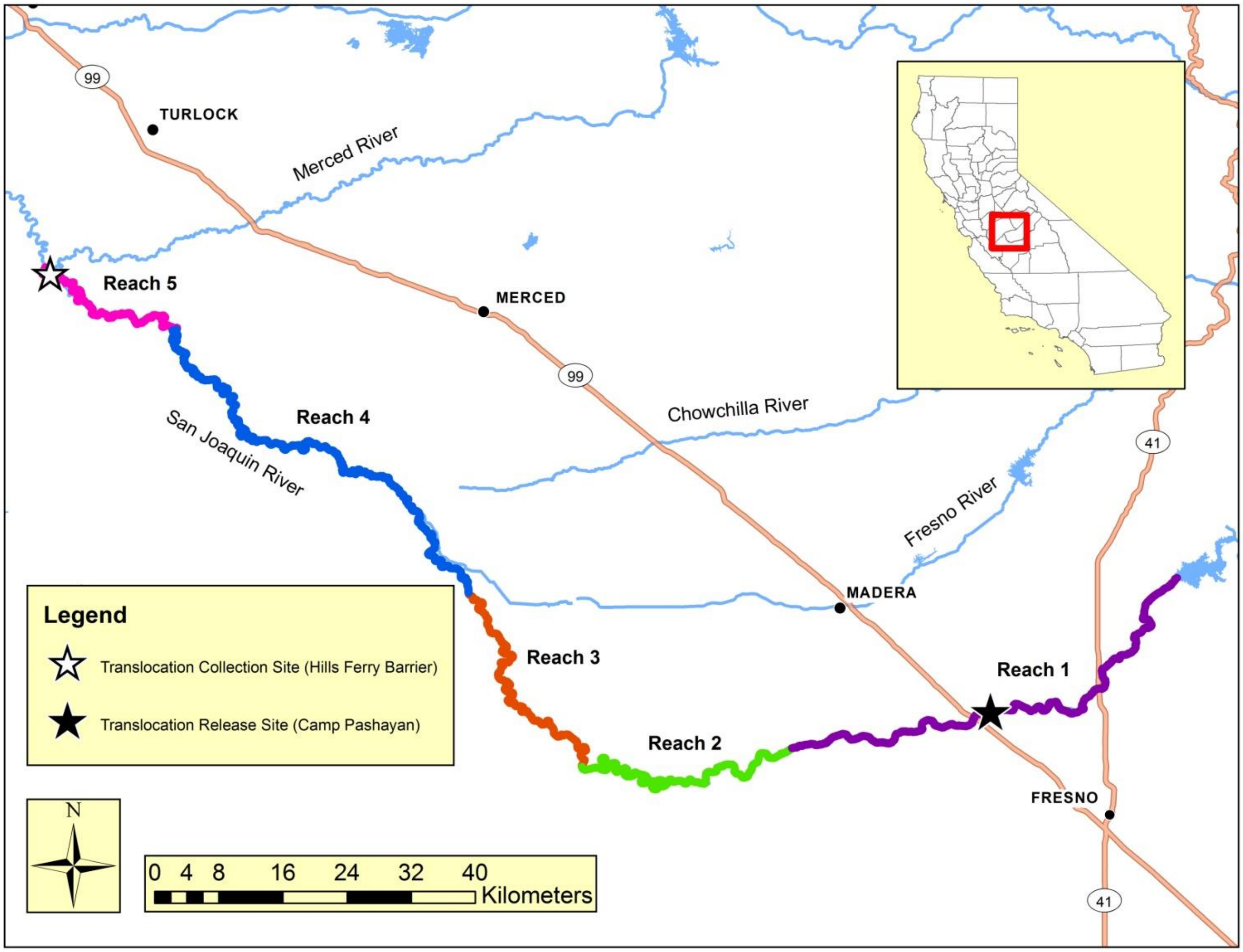

Figure 1. The San Joaquin River Restoration Area, and adult fall-run Chinook Salmon translocation collection and release locations within the San Joaquin River, CA. The Restoration Area is stratified into five reaches delineated using labels and unique colored lines. Note: adult salmon were collected during 2013 and 2014 at several locations in Reach 5 upstream of the Hills Ferry Barrier.

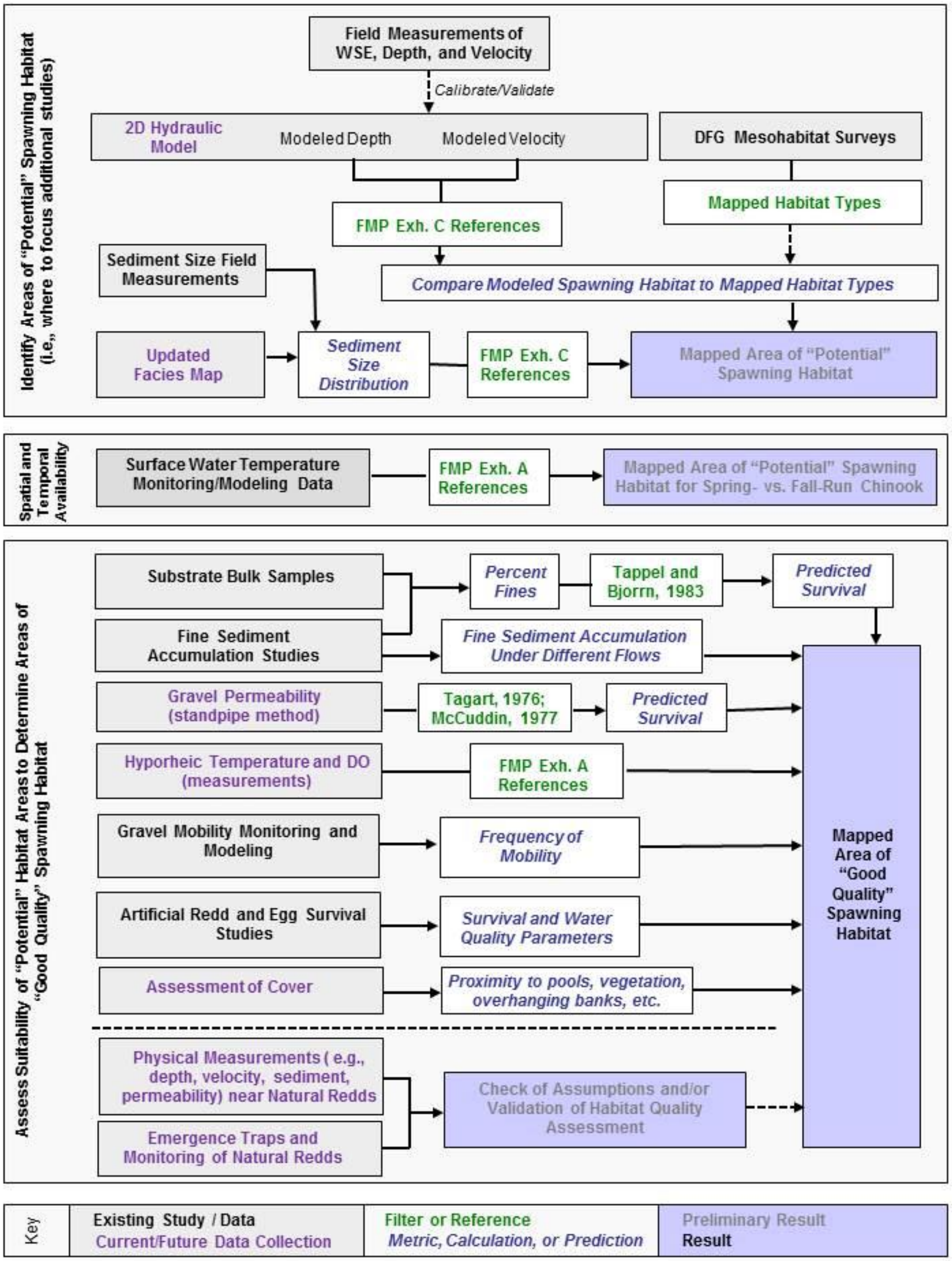

Figure 2. Example of some of the physical characteristics hypothesized to effect spawning success and how studies could provide data to assess spawning habitat availability and quality in the Restoration Area.

1.2 Methods

We assessed the spawning success of fall-run Chinook Salmon translocated from Reach 5 to Reach 1 within the Restoration Area by monitoring salmon carcasses, redds, and the emergence of fry from observed redds.

1.2.1 Study Area

The San Joaquin River basin has a Mediterranean-montane climate (i.e., wet-cool winters and dry-hot summers; Null and Viers 2013) and is subjected to excessive groundwater pumping to support agricultural land use (Galloway and Riley 1999; Traum et al. 2014). As a result, Reach 1 is dominated by losing or effluent reaches and is largely managed to ensure the flow at the bottom of Reach 1 (i.e., at Gravelly Ford) is 0.14 m^3/s during non-flooding conditions. In general, Reach 1 is dominated by continuous flows conveyed in an incised gravel-bedded channel that is confined by periodic bluffs or terraces (SJRRP 2010). In addition, the channel bed is moderately sloped, and contains off-channel and on-channel mine pits from historic sand and gravel mining operations (SJRRP 2010). Land use in areas adjacent to the study area is dominated by anthropogenic urban and agricultural developments (Traum et al. 2014).

1.2.2 Fish Translocation

The U.S. Bureau of Reclamation (USBR) and CDFW collected adult fall-run Chinook Salmon upstream of the HFB (rkm 118) in Reach 5 and translocated them to Reach 1 of the Restoration Area during the fall of 2013 and 2014. In 2013, collections occurred from October 1, 2013 to December 15, 2013. In 2014, collections occurred from November 1, 2014 to December 15, 2014. The USBR captured adult salmon using fyke nets or temporary weirs at several locations in the mainstem San Joaquin River and Mud and Salt sloughs. In addition, the CDFW captured adult salmon using dip nets at the terminal end of irrigation canals upstream of the HFB. Captured fish were sexed, measured for fork length (FL) to the nearest millimeter, assessed for level of maturation (expression of milt or eggs), checked for the presence of an adipose fin and overall condition (e.g., good or poor), and tagged using a uniquely numbered Peterson disc tag. In addition, the majority of females had an acoustic tag (i.e., VEMCO, V13, 69 kHz transmitter) implanted into their abdomen intragastrically using a balling gun to allow CDFW to monitor post-release movements. Thereafter, individuals were transported by truck in oxygenated ($\geq$ 8 mg/L) and salted (6–10ppt) tanks (1700–2300 L), and released in the San Joaquin River at Camp Pashayan near the Highway 99 bridge (rkm 391; Figure 3). In general, fish were considered in poor condition if, ,~~-~~they were lethargic (e.g., listing to one side) or possessed physical anomalies (e.g., saprolegnia, extensive fin erosion, or large lacerations).

A total of 353 and 479 adult fall-run Chinook Salmon were translocated to Reach 1 in 2013 and 2014, respectively (Table 1). Approximately 120 females were translocated during each year. However, there were 51% more males translocated in 2014 relative to 2013 (Table 1). In general, less than 7% of all fish translocated were noted to be in poor condition (Table 1). In addition, we assumed that the majority of translocated fish were of hatchery origin based on 25.7% and 26.6% of individuals had a clipped adipose fin in 2013 and 2014, respectively (Table 1). Currently, Central Valley hatcheries implement the constant fractional marking of at least 25% of hatchery produced juvenile fall-run Chinook Salmon

(Nandor et al. 2010). The majority of males and females were translocated during November of both years (Table 2).

Table 1. The number, mean size (± 95% confidence intervals), sex, condition, level of maturation, and proportion of adipose clipped adult fall-run Chinook Salmon translocated from Reach 5 to Reach 1 of the Restoration during the fall of 2013 and 2014.

Trap and Haul Metrics	2013		2014	
	Female	Male	Female	Male
Adults Translocated	116	237	121	358
Average Fork Length (mm)	761 ± 15	710 ± 15	701 ± 18	658 ± 10
Poor Condition (%)	5.2%	2.9%	6.6%	3.6%
Expression of Eggs/Milt (%)	63.8%	91.6%	62.0%	97.2%
Adipose Clipped (%)	34.5%	22.8%	22.3%	26.8%

Table 2. The number of male and female adult fall-run Chinook Salmon translocated from Reach 5 to Reach 1 by month in 2013 and 2014.

Month	2013		2014	
	Female	Male	Female	Male
October	5	19	---	---
November	57	135	72	210
December	54	83	49	148

1.2.3 Carcass and Redd Monitoring

Sampling Design.— We monitored carcasses and redds by conducting structured visual observation surveys. These surveys were implemented by a crew of three to six USFWS and CDFW staff floating the river on a drift boat and kayaks. The survey was implemented once a week after the beginning of adult translocations (e.g., November) to January, which included a total of eight survey periods (i.e., eight weeks) during both years. We conducted these surveys each week during daylight hours between Friant Dam (430 rkm) and the Highway 99 bridge (391 rkm) in 2013 and 2014. In 2014, the survey was extended further downstream to Donny Bridge (387 rkm) to ensure the majority of potential spawning habitat was monitored. In general, the sampling extent was stratified into 2–3 reaches and was sampled over a span of 2–4 d. Due to access limitations, we were unable to survey approximately 210 m immediately downstream of Friant Dam, which included a large plunge pool and a single riffle complex. Each weekly survey occurred during the middle of the week to keep a consistent number of days from the time an area was surveyed until it was revisited.

During each survey, the drift boat navigated down the river center or in the thalweg where standing crew members were best able to observe deeper water. The kayaks were used to survey side channels, river margins, and shallow areas not easily accessed by the drift boat. This allowed for complete coverage of all but the widest and deepest sections of river (e.g., in-channel mine pits). The drift boat used oars in passing through riffles, and used an outboard motor to efficiently pass through long and deep pools, which had a low likelihood of containing redds or observing carcasses.

Redd Sampling.— We identified redds by the presence of substrate cleaned of periphyton, a defined pit (depression) and tailspill (mound) within the substrate. All redds detected for the first time were assigned a unique redd identification number, had their location recorded using a GPS unit, measured (i.e., pit and tailspill length, width, and depth), and assessed for hydrogeomorphology (i.e., dominant substrate size and water velocity). We measured the dimensions of each redd's pit and tailspill using a metric tape to the nearest 0.01 m. We calculated the area of the redd by summing the area of the pit and tailspill, which was calculated by multiplying their length by their width. Depth measurements were taken pre-redd in undisturbed substrate, immediately upstream of the pit, and as close as possible to the incision of the sunken area (i.e., pit) using a top-set rod. A gravelometer was used to estimate dominant particle size of the substrate within the tailspill and immediately upstream of the redd as sand/silt (<2.8 mm), gravel (2.8–32 mm), large gravel (32–64 mm), small cobble (64–90 mm), medium cobble (90–128 mm), large cobble (128–180 mm), or boulder/bedrock (>180 mm). If a gravelometer was not available, substrate size was measured on the axis of intermediate length using a metric tape. For example, if a rock measured 2 x 4 x 3 cm, then we estimated the particle size to be 3 cm. Lastly, we measured the pre-redd mean water column velocity at 60% depth when water depth was ≤1 m or 20% and 80% depth when water depth was >1 m using a velocity meter (e.g., OTT MF Pro or Marsh McBirney 2000) in conjunction with a top-set rod.

The number of redds affected by superimposition were recorded each year. We defined superimposition as the presence of overlap between two or more redds. In 2014, we installed redd grates on all redds as part of a redd superimposition study intended to evaluate the effectiveness of redd grates in minimizing redd superimposition within and among salmon runs within the Restoration Area. We placed redd grates over the majority of the tailspill area and anchored them to the substrate using cobble. Redd grates were constructed by using 3-foot pieces of reinforcing bar connected with plastic ties to form triangles.

Carcass Sampling.— All adult carcasses detected were assigned a unique identification number and were assessed for the presence of an adipose fin, disc tag, and acoustic tag, and status of decomposition. If possible, carcasses had their otoliths removed and preserved for future analyses (e.g., aging, growth, microchemistry). We identified all carcasses with a clipped or missing adipose fin as carcasses with a CWT, assumed they were of hatchery origin, and their heads were taken, bagged, and returned to the Lodi Fish and Wildlife Office for CWT extraction and reading to determine their origin. We determined the origin of these individuals by obtaining tag information (race and hatchery origin) from the Regional Mark Information System maintained by the Pacific States Marine Fisheries Commission. Decomposition was determined by the presence of clear eyes or blood remaining within the gills. Carcasses that possessed one clear eye or pink coloration within the gills were considered “fresh,” otherwise carcasses were considered “decayed” or a “skeleton” pending the exposure of bones (i.e., missing flesh). All fresh or partially decayed carcasses were evaluated for sex (male or female), retention of eggs in females (spawned, few or no eggs are present; partially spawned, some eggs present; or unspawned, many eggs are present), measured to the nearest millimeter FL, and scale samples collected. We determined the sex of individuals by dissecting their abdominal cavity and looking for the presence of testes or ovaries. We examined the abdominal cavity for the presence of an acoustic tag if the carcass was identified as a female to be able to recycle the tag for further use. Carcasses in the advanced stages of decomposition were given an unknown designation for sex. Although scale and otolith samples were part of these surveys, they were collected for research beyond the scope of this report and will not be discussed further.

In 2014, fresh carcasses were marked using one or two external tags (e.g., individually numbered aluminum tags attached by hog ring to their maxilla) post processing to quantify escapement. To accommodate external marking, fresh carcasses with a clipped or missing adipose fin did not have their heads removed for CWT extraction during 2014. Escapement is defined as the number of individuals that escaped the recreational and commercial fisheries (i.e., survived) and were capable of producing offspring (Ross 1997). Although there is no commercial or recreational fishing permitted in the Restoration Area, there is a possibility of poaching. Unique tag codes were used for each individual to determine what week an individual was originally detected. Once we marked fresh carcasses post processing, we released them in flowing water to ensure "mixture" of the marked population. If marked carcasses were recaptured in subsequent weeks, we identified them as a recapture and their tag codes were recorded. After processing unmarked carcasses designated as decayed or skeletons or marked carcasses that are recaptures, we cut the tail off of the carcass (between adipose and caudal fin) to prevent the carcass from being double counted.

Analysis.— Redd and carcass results were related to environmental variables hypothesized to influence adult survival or spawning success of Chinook Salmon translocated into Reach 1 (SJRRP 2010). In particular, we were interested in comparing flow and water temperature within and among survey years to the abundance and distribution of redds observed during 2013 and 2014. We obtained mean daily water temperature and mean daily flow data from the California Data Exchange Center (http://cdec.water.ca.gov) at three gaging stations distributed throughout the study area. These gaging stations were located at Donny Bridge (387 rkm), at the Highway 41 bridge (410 rkm), and near Friant Dam (428 rkm). We estimated the spawning success of translocated females by dividing the total number of redds observed by the total number of females translocated during each field season.

To help determine the number of adult salmon available to spawning successfully, we estimated salmon escapement in 2014 using a modified Cormack-Jolly-Seber (CJS) mark-recapture maximum likelihood estimator (following Bergman et al. 2012). The primary assumptions of the CJS estimator were that (1) marks were retained and identified correctly, (2) survey periods were instantaneous (i.e., no births or deaths during the weekly survey), and (3) that emigration from the population was permanent (Bergman et al. 2012). We evaluated a total of nine candidate CJS models that represented competing hypotheses concerning how the sex and length of carcasses influenced the detection or survival probabilities. We fit each candidate model using a bootstrapping procedure and 5,000 replicates. The best approximating and most parsimonious model was used to estimate the number of adult translocated salmon available to spawn in the Restoration Area. We used an information theoretic approach (Burnham and Anderson 2002) and determined the best approximating candidate model as that with the lowest Akaike Information Criterion (Akaike 1973) with small sample bias adjustment (AIC_c; Hurvich and Tsai 1989). To assess the amount of evidence one candidate model had over another, we calculated the ratios of Akaike weights (w_i; Burnham and Anderson 2002). Any candidate model with Akaike weights that were within 12% of the best-approximating candidate model's Akaike weight was included within the confidence set of models (following Royall 1997).

1.2.4 Emergence Monitoring

Sampling Design.— We conducted the emergence monitoring in conjunction with the carcass and redd monitoring. We monitored the emergence of fry from redds detected during the carcass and redd ~~monitoring~~ surveys using emergence traps (Koski 1966; Hausle and Coble 1976; Beacham and Murray 1985; TID and MID 1991). A total of five and ten redds were selected for emergence monitoring among the redds detected during the carcass and redd monitoring in 2013 and 2014, respectively. In general, only redds that were distinct, not superimposed on or under other redds, readily accessible by foot, and those capable of having their entire egg pocket covered by an emergence trap were considered for monitoring. In 2013, we considered redds for monitoring that were detected prior to December 15, 2013 and those located near previous artificial egg study sites. The artificial egg study sites were originally established in 2011 to assess how egg survival varied among water quality and physical conditions occurring in our study area (SJRRP 2012b). In 2014, we considered redds for monitoring that were detected throughout the entire carcass and redd survey extent in space and time.

Prior to installing emergence traps, we collected daily water temperature data for each redd at the nearest California Data Exchange Center flow gage station (Donnie Bridge, 387 rkm; Highway 41 Bridge, 410 rkm; near Friant Dam, 428 rkm) to estimate emergence timing using accumulated thermal units (ATUs; Beachum and Murray 1990). We calculated ATUs by adding average daily temperatures, 1 ATU = 1 °C for 1 day (Berejikian 2011). In general, we assumed that emergence started at approximately 700ATU. We installed emergence traps 3–14 days prior to the start of assumed emergence. We installed emergence traps on selected redds no more than two weeks prior to the start of expected emergence to minimize the potential for the traps to influence the hydrogeomorphology within monitored redds and ensure we would collect all emerging fry.

Figure 3. An emergence trap installed in the San Joaquin River at rkm 421 in 2014.

Emergence Trap Sampling.— The emergence traps consisted of 0.32 cm nylon mesh covering a steel frame and a canvas skirt made of 30.48 cm Dacron sailcloth, which was buried straight down into the gravel to a depth of approximately 30.48cm to minimize lateral escapement of fish. Each emergence trap was tear-shaped and contained a live-box at the narrower caudal end of each trap, which was oriented downstream. The emergence traps measured 2.42 m long and 1.83 m at their widest point; covering approximately 2.83 m^2. The livebox was assembled using a 3.79 L wide-mouth polyethylene bottle and a 15 cm diameter funnel to collect emerging fry. The bottom of the bottle was cut out and the funnel attached with silicone. We cut holes into both sides of the bottle and 0.32 cm polypropylene mesh was attached with silicone to create a vent to allow water to escape and minimize fish mortality. A sock was constructed of Dacron sailcloth extended from the opening of the back to the holding bottle and was attached using a hemmed drawstring which grabbed the lip of the funnel.

During installation, each emergence trap was carried over to the selected redd and placed on top of the distinct egg pocket(s). Subsequently, we installed rebar measuring 0.95 cm thick by 76.20 cm long around the frame and secured it to the riverbed using washers and hose clamps. Thereafter, a trench was excavated around the edges of the trap at a depth of 30.48 cm or until the substrate became too armored where digging could no longer continue. Finally, we buried the canvas skirt within the trench, backfilled the excavated area using materials excavated earlier, and attached the live-box to the narrow caudal end of the emergence trap.

Emergence traps were checked and cleaned 2–4 times each week. However, we checked the emergence traps daily during periods of peak emergence. Emerged alevin or fry captured within the live-box were counted and measured to the nearest millimeter FL. In 2014, other fish species (e.g., cottids) were identified to species, measured, and enumerated. After processing, we released all fish to the river. When each emergence trap was removed at the end of the season when fish were no longer being collected, we assessed the redd for unviable eggs, entombed alevins, and emerged juveniles that did not enter the live-box to better assess survival rates or identify the presence of a “false redd” (i.e., no eggs were ever deposited at the location).

Analysis.— Emergence results were related to environmental variables hypothesized to influence the survival of egg to emerging fry within Reach 1 (SJRRP 2010). In particular, we were interested in relating flow, water temperature, and hydrogeomorphology (i.e., redd depth and substrate composition). within and among survey years to the number of emerged fry observed in each monitored redd. We obtained mean daily water temperatures and mean daily flow from the California Data Exchange Center (http://cdec.water.ca.gov) at three gaging stations evenly distributed throughout the study area. These gaging stations were located at Donny Bridge (387 rkm), at the Highway 41 bridge (410 rkm), and near Friant Dam (428 rkm). We assigned emergence traps to one of three reaches based on their proximity to the gaging stations: Donny Bridge Reach (386–397rkm), Highway 41 Reach (397–417rkm), and Friant Dam Reach (417–429rkm). The redd depth and substrate composition data were obtained from the data collected as part of the redd sampling described earlier.

To evaluate the number of fry produced from each spawned female, we averaged the number of fry that emerged from monitored redds. The primary assumptions of this approach were that (1) the emergence traps are 100% efficient, (2) each emergence trap covered all of the eggs deposited by a single female, (3) emergence counts were representative across all observed redds, and (4) the fishes within each emergence trap exhibited a closed population (i.e., no fish can immigrate into the emergence traps). We also examined the relative importance and effect of environmental variables (e.g., flow, temperature, substrate, depth) on fry emergence count using generalized linear models assuming a Poisson distribution (Williams et al. 2002). We used an information theoretic approach (Burnham and Anderson 2002) to evaluate the relative fit of 11 candidate models representing a variety of *a priori* hypotheses regarding the influence of the environmental factors on egg survival or emergence count. In particular, we were interested in evaluating the influence of six predictor variables on fry emergence count: year, reach, maximum 10-day temperature during incubation, mean daily temperature during incubation, mean daily flow during incubation, dominant substrate type at time of detection, and pre-pit water velocity at time of detection. We calculated Pearson correlation coefficients for all pairs of predictor variables and none were considered highly correlated (r^2 >0.5; Dormann et al. 2013). All continuous data were standardized with a mean of zero and standard deviation of one to facilitate model fitting. We created binary indicator variables for year (baseline = 2013), reaches (baseline = Donny Bridge Reach) and dominant substrate type (baseline = gravel). Because flow and year were highly correlated, they were not included within the same candidate models. The goodness-of-fit was evaluated by graphing the residuals against predicted values for the best approximating model.

1.3 Results and Discussion

The San Joaquin River basin experienced the first and second year of a critical drought in 2013 and 2014, respectively. In general, river flow decreased from Friant Dam to Donny Bridge, and varied among and within years (Figure 4). River flow ranged from 3.19–19.7 m³/s during 2013, whereas flow ranged from 3.48–6.86 m³/s during 2014. Water temperatures were, on average, cooler further downstream from November to January during both years presumably due to cooler ambient air cooling Friant release water (Figure 5). However, water temperatures were considerably warmer during the fall (October–November) of 2014 relative to the fall of 2013. During both years, water temperatures remained below the upper lethal spawning temperature limit (17°C) from December to February (Figure 5).

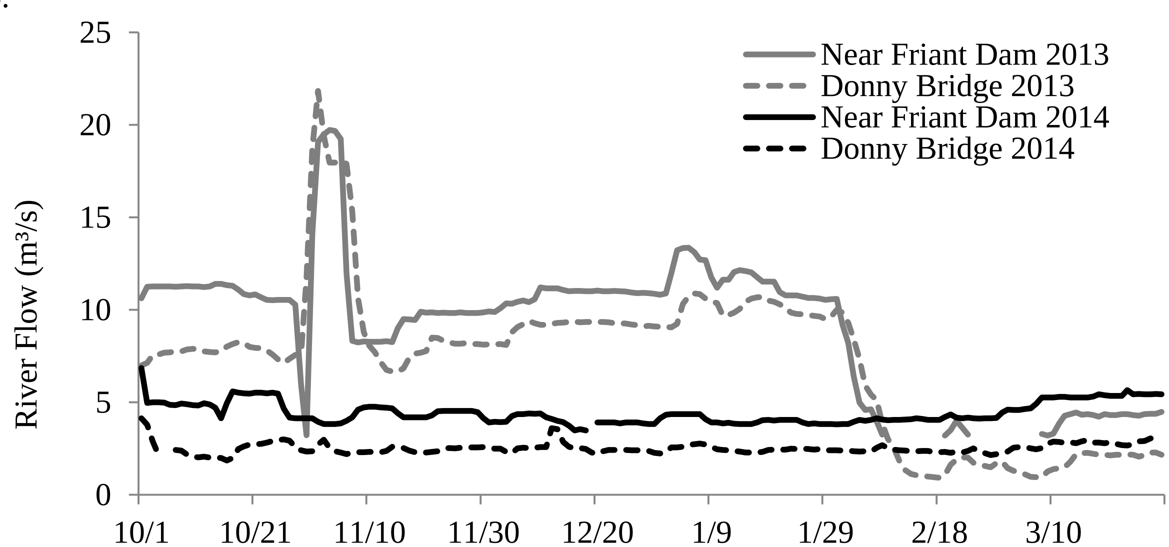

Figure 4. Mean daily river flow (m^3/s) near Friant Dam (rkm 430) and at Donny Bridge (rkm 386.9) within the study area from October to March during 2013 and 2014. Data were obtained from the California Data Exchange Center (http://cdec/water/ca/gov).

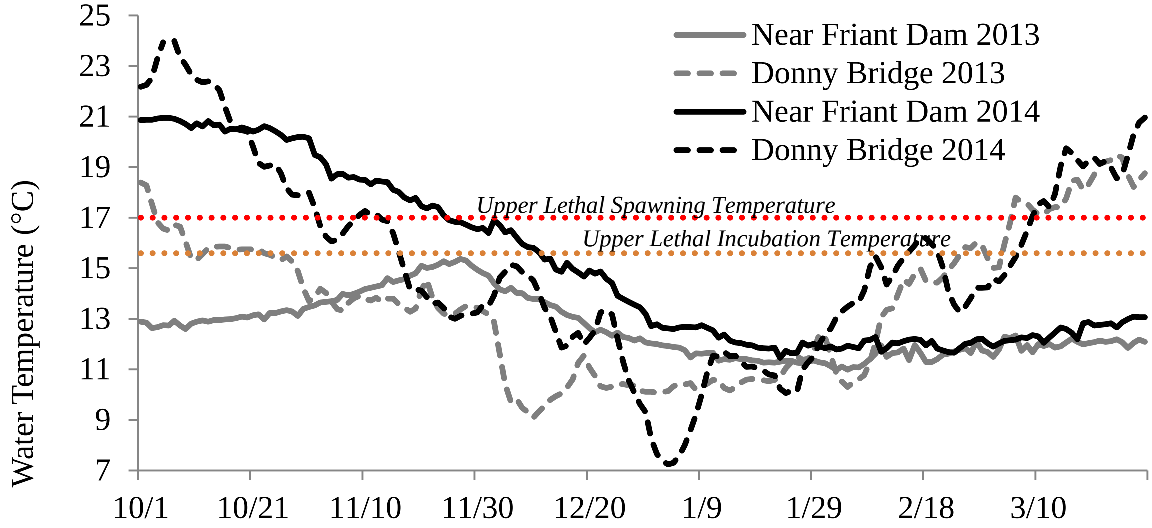

Figure 5. Mean daily water temperature (°C) near Friant Dam (rkm 430) and at Donny Bridge (rkm 386.9) within the study area from October to March during 2013 and 2014. Data were obtained from the California Data Exchange Center (http://cdec/water/ca/gov). The upper lethal threshold for Chinook Salmon during spawning and incubation is 17^{o}C and 15.6^{o}C, respectively (SJRRP 2010).

1.3.1 Carcass and Redd Monitoring

Redd Sampling.— We observed a total of 73 and 81 redds in 2013 and 2014, respectively (Figure 6). As a result, we estimated the spawning success of translocated females to be 63% in 2013 and 67% in 2014. In 2013, 5 of 73 redds were detected below the Highway 99 bridge while staff implemented another study, which provided evidence that we needed to extend the redd and carcass survey further downstream during the 2014 field season.

The temporal distribution of redds were similar among years. Redds were detected from November to the beginning of January (Figure 6), which is likely due to the timing of fish translocation (Table 2) coupled with natural fall-run spawn timing. The number of detected redds peaked during the first or second week of December (Figure 6) when water temperatures were below the upper lethal spawning temperature limit (17°C; Figure 5). In response to the reduction in flow in February during the 2013 field season (Figure 4), we implemented a two-day survey (February 20–21, 2014) to determine if redds had been dewatered. We determined that approximately 15% (n=11) of the redds were at least partially dewatered (defined as at least a portion of the redd's tailspill was not submerged) in 2013. However, it is possible that more redds were dewatered than the results indicate because staff were unable to conduct the survey at the period when flows were the lowest (February 14, 2014).

The spatial distribution of redds varied among years. In 2013, redds were more evenly distributed throughout the study area (Figure 7). Whereas the majority of redds were detected closer to the Translocation Release Site (Camp Pashayan) in 2014 (Figure 8). This may be based on the presence of less river flow and warmer water temperatures in 2014 relative to 2013 (Figures 4 and 5). In addition, we observed a total of 10 redds (14%) being influenced by superimposition in 2014. We hypothesize that the presence of superimposition in 2014 was based on the higher redd densities observed near the Translocation Release Site coupled with less spawning habitat suitability or availability. Redd area was measured to be, on average, 10.2m^2 and 11.6m^2 in 2013 and 2014, respectively. Redds occurred at depths of 0.2–1.0 m and in habitats containing mean water velocities of 0.2–1.2 m/s that were dominated by gravel (2.8–32 mm) or large gravel (32–64 mm; Table 3), which is consistent with previous studies (Williams 2006 and references therein).

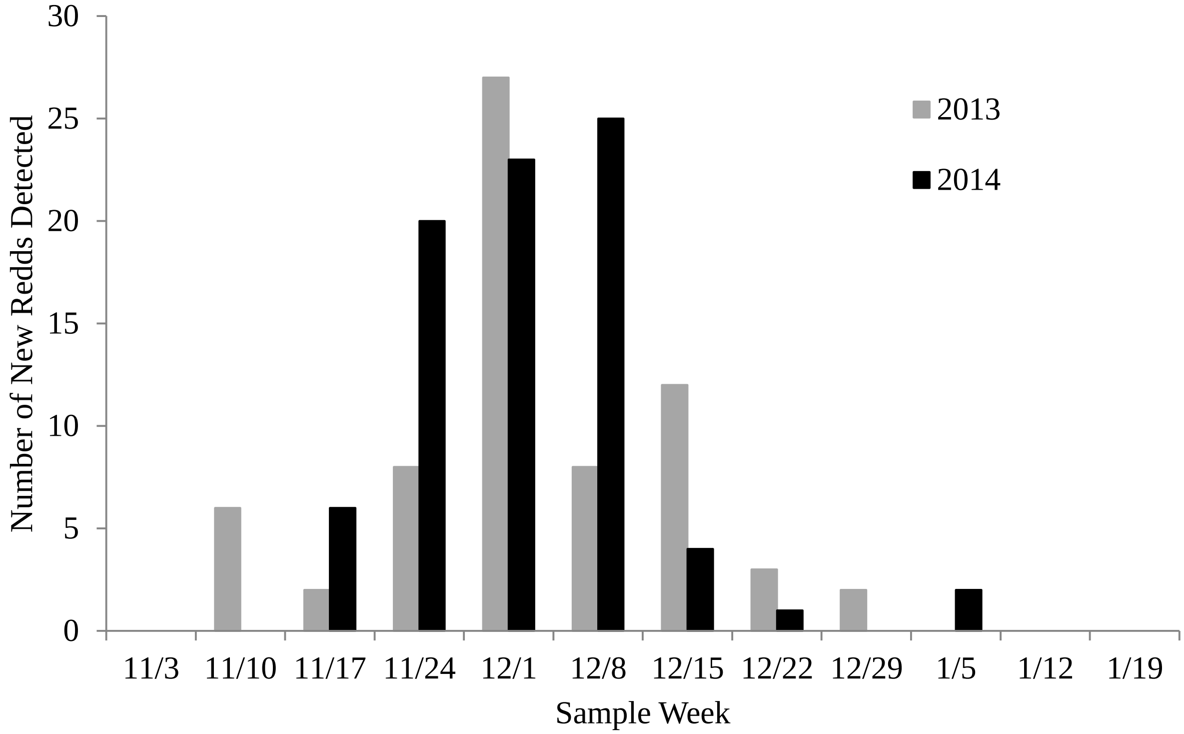

Figure 6. The number of redds detected for the first occasion within Reach 1 of the San Joaquin River by sample week during the 2013 and 2014 field seasons.

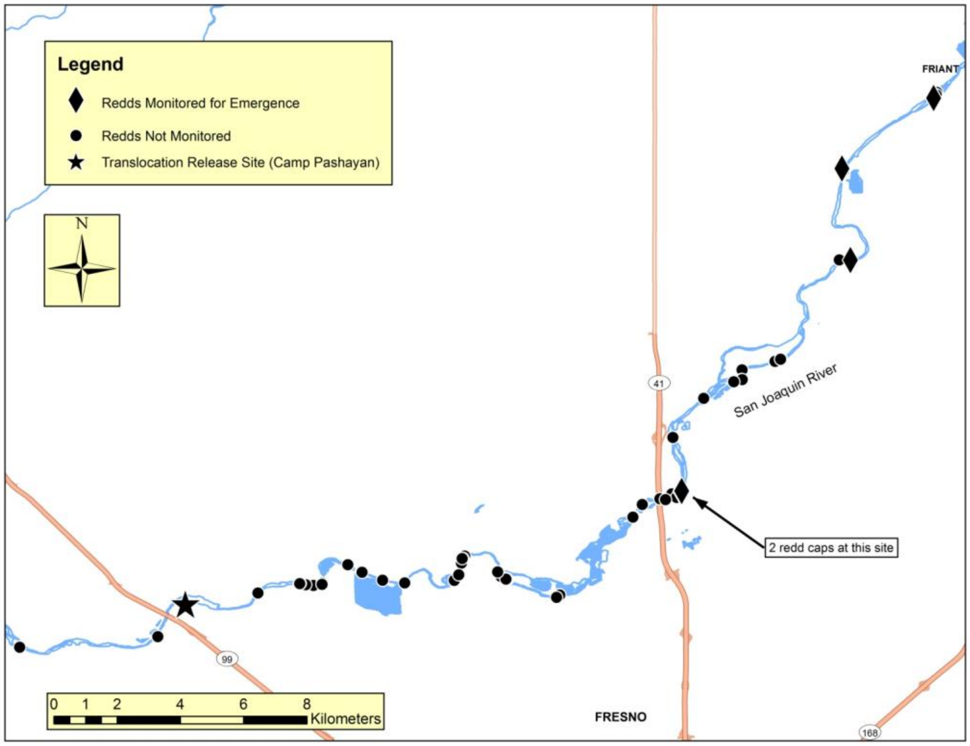

Figure 7. The locations of redds detected and the subset of redds monitored for emergence within Reach 1 of the San Joaquin River, CA during 2013.

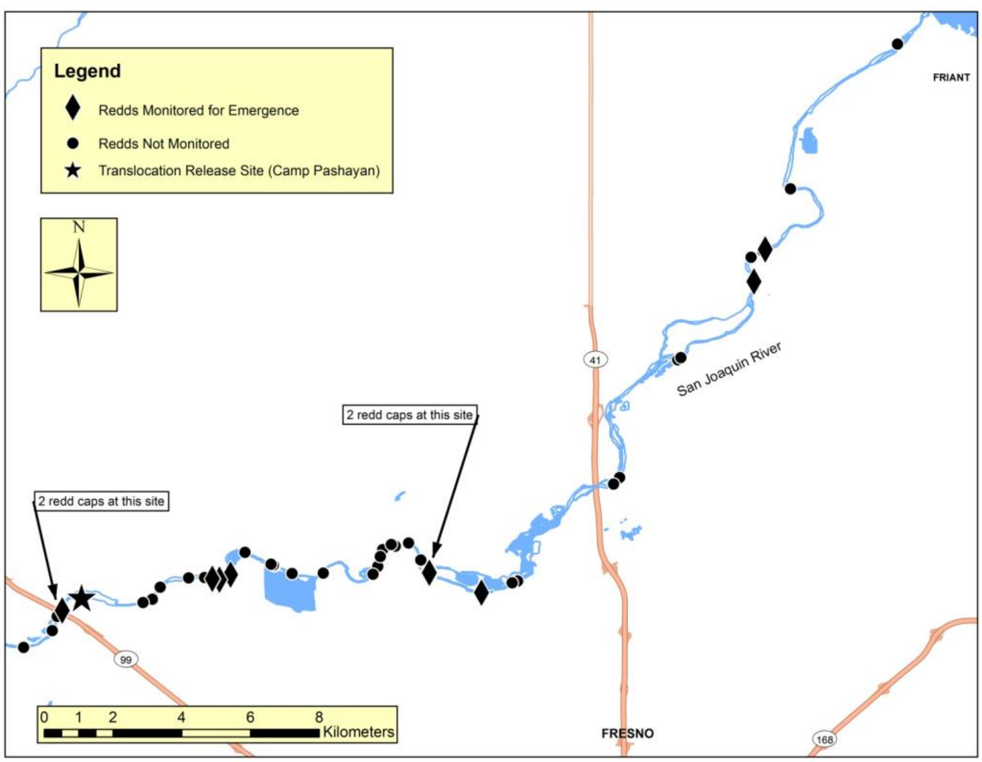

Figure 8. The locations of redds detected and the subset of redds monitored for emergence within Reach 1 of the San Joaquin River, CA during 2014.

Table 3. Mean, standard deviation (in parentheses), and range of redd and habitat characteristics in observed redds within Reach 1 of the San Joaquin River Restoration Area during the 2013 and 2014 field seasons.

Variable	2013		2014	
	Mean (SD)	Range	Mean (SD)	Range
Total Redd Area (m²)	10.23 (7.30)	2.64–28.0	11.63 (7.9)	0.88–38.28
Redd Tailspill Area			8.56 (6.0)	0.64–29.64
Redd Pit Area			3.02 (2.58)	0.14–11.56
Maximum Pit Depth (m)Depth	0.57 (0.15)	0.30–1.02	0.51 (0.14)	0.22–0.98
Depth Upstream of Redd (m)	0.47 (0.16)	0.20–0.96	0.41 (0.15)	0.10–0.90
Mean Velocity Upstream of Redd (m/s)	0.72 (0.25)	0.22–1.45	0.61 (0.19)	0.24–1.15

Table 4. The dominant substrate type immediately upstream of observed redds within Reach 1 of the San Joaquin River Restoration Area during the 2013 and 2014 field seasons.

Dominant Substrate (%)	2013	2014
Sand/silt (<2 mm)	0%	0%
Gravel (2–32 mm)	31%	30%
Large gravel (32–64 mm)	48%	70%
Small cobble (64–90 mm)	19%	0%
Medium cobble (90–128 mm)	0%	0%
Large cobble (128–180 mm)	2%	0%

Carcass Sampling.— A total of 33 and 98 Chinook Salmon carcasses were detected during the 2013 and 2014 field seasons, respectively. Approximately 44% of the carcasses processed were female during both years (Table 5). In 2013, the majority (71%) of recovered females were classified as fully spawned. In contrast, less than 50% of recovered females were classified as fully spawned in 2014 (Table 5). We also found evidence that the residency time of individuals (defined as the time between translocation release and carcass detection) was, on average, shorter in duration in 2014 relative to 2013 (Figure 9), which is consistent with the truncated spatial distribution of redds observed in 2014.

In general, carcass length and the proportion of carcasses containing an adipose fin clip were representative of the salmon translocated each year (Tables 2 and 5). During the redd and carcass surveys, we recovered a total of 6 and 21 carcasses during the 2013 and 2014 field season, respectively, that had an adipose fin clip and contained a CWT (Table 5). In addition, 9 and 3 carcasses containing a CWT were obtained from other SJRRP field activities in 2013 and 2014, respectively. The majority (≥70%) of carcasses containing a CWT were of Mokelumne River Hatchery origin during both years (Figure 10) and were primarily 3 years of age (Figure 11). However, there was a higher proportion of age-2 individuals in 2014 relative to 2013.

In 2014, a total of 55 fresh salmon carcasses were tagged and included in our mark-recapture study to estimate the total number of salmon available for spawning (i.e., escapement). Although the best approximating CJS candidate model was one that indicated that survival and detection probabilities were positively related to fish length, the fit was equivalent to the most parsimonious candidate model that assumed that detection and survival was constant among sex and fish length. As a result, we used the most parsimonious CJS model to estimate escapement. The model estimated that our capture probability was, on average, 38.5% and that a total of 189 (150–282; 95% confidence limits) adult translocated salmon were available for spawning in 2014. These results suggest that translocated salmon may have been poached in Reach 1 or traveled outside of our sampling extent. Two salmon carcasses detected in 2014 showed signs of poaching including angling injuries. Assuming that 44% of the adults were females (Table 5), we projected that a total of 87 translocated females had the potential to spawn (i.e., escape), which is consistent with the number of redds (n= 81) detected in 2014. Therefore, we are confident that pre-spawn mortality was higher than the 15% threshold target (SJRRP 2010).

Table 5. The number, mean size (± 95% confidence intervals), sex, spawning disposition, and proportion of adipose clipped adult fall-run Chinook Salmon recovered as carcasses and processed in Reach 1 of the Restoration Area during the 2013 and 2014 field seasons.

Carcass Recovery Metrics	2013		2014	
	Female	Male	Female	Male
Carcasses Recovered	14	18	42	54
Average Fork Length (mm)	729 ± 75	776 ± 56	738 ± 27	708 ± 33
Fish Fully Spawned (%)	71.40%	---	47.60%	---
Adipose Clipped (%)	21.40%	16.70%	21.40%	22.20%
Fish included in Mark Recapture Study	---	---	26	29

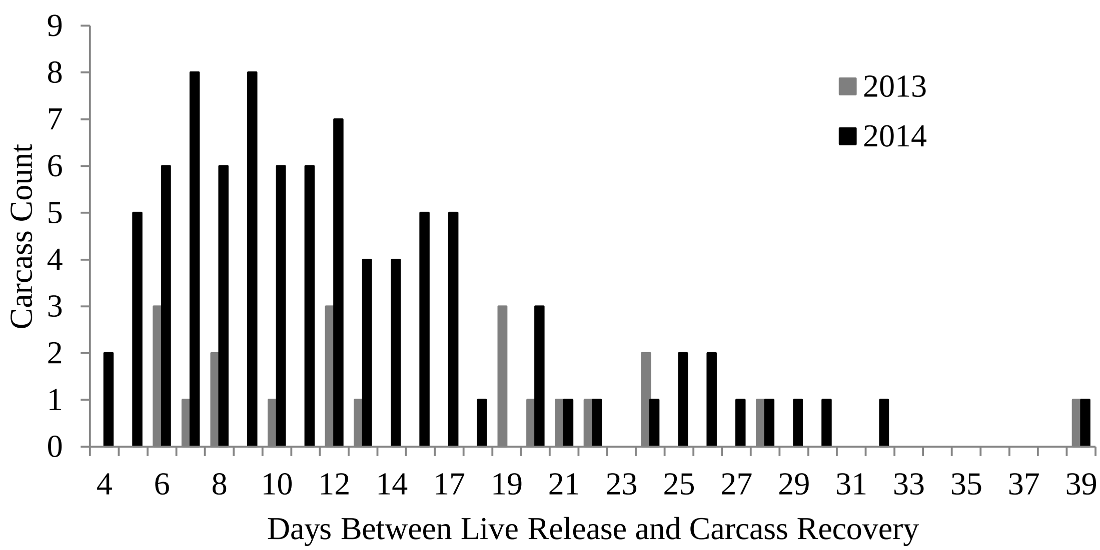

Figure 9. The time (days) between the recovery of carcasses marked with a disc tag and the live release of the individual within the San Joaquin River during the 2014 and 2015 field seasons.

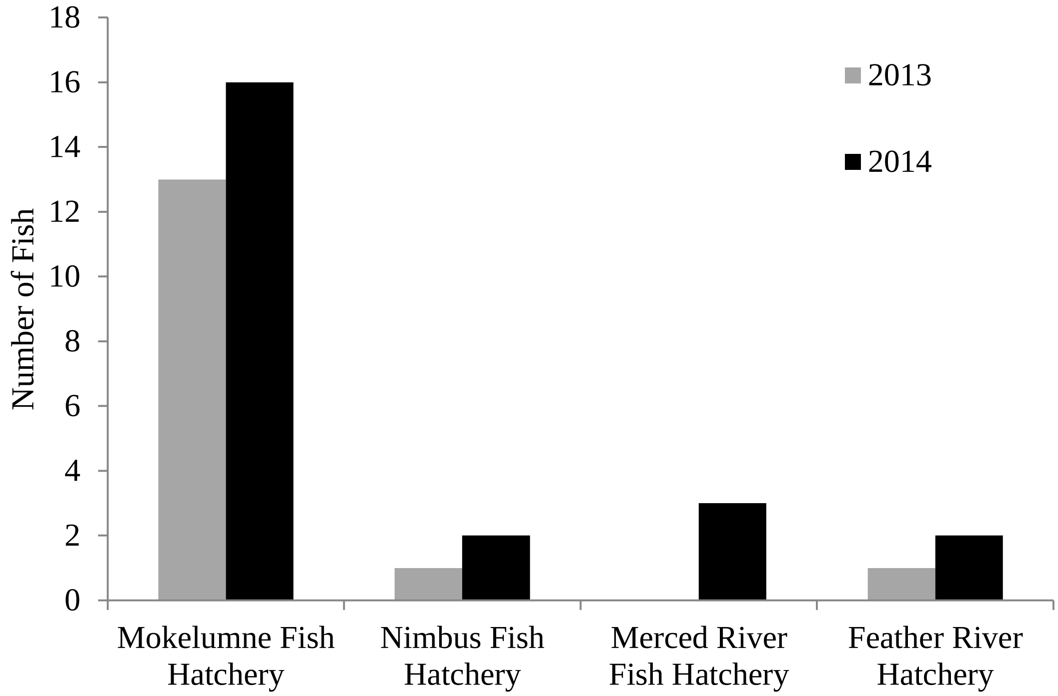

Figure 10. Hatchery origin of translocated Chinook Salmon using coded-wire tag (CWT) recovery data collected during the 2013 and 2014 field seasons. CWT data represent all carcasses collected during the carcass survey and during translocation activities.

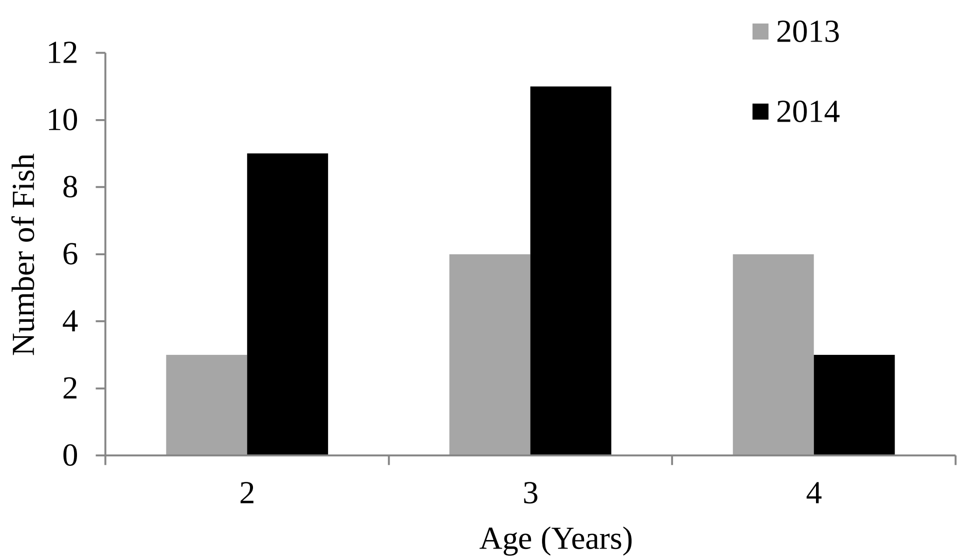

Figure 11. Age distribution of translocated Chinook Salmon using coded-wire tag (CWT) recovery data collected during the 2013 and 2014 field seasons. CWT data represent all carcasses collected during the carcass survey and during translocation activities.

1.3.2 Emergence Monitoring

A total of 15 redds were monitored for fry emergence during the 2013 and 2014 field seasons (Figures 7 and 8). We observed, on average, 1,277 salmon fry emerge from each redd monitored in 2013 (Table 6). In contrast, we observed an average of 997 salmon fry emerge from each redd monitored in 2014. Our emergence counts were lower than what was observed from 24 natural redds monitored within the lower Tuolumne River prior to any restoration (TID and MID 1991). Salmon fry emerged faster within the redds monitored during 2014 relative to those monitored in 2013 (Table 6). The faster emergence rate may be based on the presence of warmer water temperatures (Olson and Foster 1957; Boles 1988; Geist et al. 2006). Emerged fry possessed similar mean fork lengths between years (2013 = 34mm FL, 2014 = 35mm FL; Table 6).

Among the redds monitored in 2013, one redd was superimposed with another redd based on producing over 5,000 fry and having two female acoustic tags deposited within its tail spill, which was determined after the redd was capped. Further, one redd was dewatered during the decrease in flow during February (Table 6). It was uncertain if flow would increase after the dewatered redd was detected, thus we excavated the dewatered redd's emergence trap to recover any fry that were trapped. During excavation, we recovered 275 fry from the dewatered redd along with several undeveloped eggs. We did not include these individuals in our emergence counts because the disposition of these fry without our intervention is uncertain. In addition, one redd in 2013 and two redds in 2014 may have been "false redds" because no eggs or fry were observed throughout the survey period or during the emergence trap excavation at the end of the survey period. Although we do not know why these females built redds and never evacuated eggs (e.g., lack of available males, poaching, etc.), we do not believe these redds biased our results because these redds would normally be enumerated as single and legitimate or “true” redds during our redd surveys.

Environmental characteristics varied between monitored redds, reaches, and years (Table 7). In general, redds monitored for emergence were associated with cooler water temperatures, higher flow, and coarser substrate within the Friant Dam Reach relative to other reaches. Additionally, we observed that redds monitored in 2013 were exposed to cooler water temperatures, higher flows, and higher water velocities relative to redds monitored in 2014 (Table 7).

Our results suggest that emergence counts were, on average, below the SJRRP target (i.e., 50% of average fecundity or 2,100 fry) among all reaches during our two-year study. This may be due to environmental conditions (see model results below) or the condition of translocated fish. For example, the majority of translocated adults were hatchery origin strays from other tributaries and hatchery strays can be less viable relative to wild stocks in their natal range (Lindley et al. 2007). Further, each of the translocated females had an acoustic tag implanted into their abdomen intragastrically and more than half (52%) of recovered females were classified as partially spawned or unspawned in 2014. Acoustic tags could negatively affect egg evacuation among female salmonids (Berejikain et al. 2007).

To assess the relative importance of environmental variables (e.g., flow, temperature, substrate, and depth) on fry emergence count, we fit a total of 11 candidate models representing different hypotheses. The best approximating candidate model for predicting fry emergence contained reach, year, and mean water temperature as predictors and there was no support for any other candidate model (Table 8). The best approximating candidate model indicated that emergence counts increased as the mean daily water temperature approached optimum (13°C; USEPA 2003; SJRRP 2010) for Chinook Salmon spawning and incubation (Figure 12). The best approximating candidate model also indicated that emergence counts varied among reaches (space) and years (time). Emergence counts were, on average, 2.24 times higher in redds monitored during 2013 relative to those monitored during 2014 (Figure 13). This temporal variation may be based on the presence of superimposition in 2014 (SJRRP 2010) or differences in flow and temperature regimes beyond differences in maximum 10-day temperature, mean daily water temperature, and mean daily flow during incubation. Parameter estimates also indicated that emergence counts were significantly higher within Friant Dam Reach redds relative to redds in other reaches (Figure 14). This spatial variation may be based on differences in hydrogeomorphology beyond our dominant substrate type and pre-pit water velocity at time of detection. For example, our dominant substrate data may not have accurately represented the true composition of substrate (e.g., presence of sand or silt) or substrate permeability, which are known to influence the survival of salmon eggs to emergence (Tappel and Bjornn 1983; Chapman 1988; Reiser and White 1988).

Table 6. Summary of fry emergence counts, fry size, and the start and duration of emergence among redds monitored within the San Joaquin River during the 2013 and 2014 field seasons.

Year	Redd #	Location (rkm)	Emergence Start (ATU)	Emergence End (ATU)	Days of Emergence	Fry Emergence Count	Fry Size (mm)	
							Mean (SD)	Range
2013	NR05FR13	426	936	1695	66	333	35 (1.9)	29–42
2013	NR07FR13[a]	412	---	---	0	0	---	---
2013	NR08FR13[b]	412	832	1802	68	5160	34 (1.8)	28–48
2013	NR14FR13[c]	430	925	925	1	1	34 (0)	---
2013	NR15FR13	422	954	1706	64	892	37 (2.0)	29–42
2014	NR04FR14[a]	406	---	---	0	0	---	---
2014	NR08FR14	396	762	1290	40	1404	33 (1.0)	26–39
2014	NR09FR14	421	736	1118	31	1005	36 (2.2)	27–40
2014	NR21FR14	396	938	1644	50	3036	34 (1.5)	30–44
2014	NR32FR14	404	636	1427	58	617	35 (2.6)	27–47
2014	NR44FR14	396	804	1524	51	2630	36 (1.6)	32–50
2014	NR46FR14	391	723	1149	28	26	35 (1.6)	30–38
2014	NR47FR14	391	658	1297	42	894	32 (1.9)	27–38
2014	NR78FR14[a]	404	---	---	0	0	---	---
2014	NR81FR14	420	671	1047	33	358	32 (1.5)	26–35

[a] Indicates no eggs found during excavation

[b] Indicates the redd was superimposed

[c] Indicates that the redd was dewatered

Table 7. Mean, standard deviation (in parentheses), and range of environmental characteristics among redds monitored for emergence by year and reach within the Reach 1 of the San Joaquin River Restoration Area.

Year	Reach	Mean Daily Temperature (°C)		Maximum 10-Day Temperature (°C)		Mean Daily Flow (m^3/s)		Mean Water Velocity (m/s)		Dominant Substrate Type	
		Mean (SD)	Range	Mean (SD)	Range	Mean (SD)	Range	Mean (SD)	Range	Gravel %	Small Cobble %
2013	Donny Bdg	12.6 (0)	---	14.6 (0)	---	10.2 (0)	---	0.8 (0)	---	100	0
n=5	Highway 41	13.2 (0.3)	13.1–13.4	15.9 (0.1)	15.8–15.9	7.6 (0)	---	1.0 (0.4)	0.7–1.2	0	100
	Friant Dam	12.3 (0.1)	12.2–12.3	14.6 (0.7)	14.1–15.0	7.6 (0.1)	7.6–7.7	0.8 (0.1)	0.7–0.9	50	50
	All	12.7 (0.5)	12.2–13.4	15.1 (0.8)	14.1–15.9	8.1 (1.2)	7.6–10.2	0.9 (0.2)	0.7–1.2	40	60
2014	Donny Bdg	12.8 (0)	---	14.7 (0)	---	4.1 (0)	---	0.4 (0.1)	0.3–0.5	100	0
n=10	Highway 41	13.3 (0.2)	13.1–13.5	14.9 (0.7)	13.7–15.7	4.2 (0.1)	4.0–4.3	0.7 (0.1)	0.5–0.9	33	77
	Friant Dam	12.9 (0.6)	12.4–13.3	14.5 (2.6)	12.7–16.4	4.3 (0.3)	4.1–4.5	0.6 (0.02)	0.62–0.65	0	100
	All	13.1 (0.3)	12.4–13.4	14.8 (1.0)	12.7–16.4	4.2 (0.1)	4.0–4.5	0.6 (0.2)	0.3–0.9	40	60

Table 8. Predictor variables, number of parameters (K), AIC_c, ΔAIC_c, and Akaike weights (W_i) for the set of candidate models predicting the number of salmon emerging from redds monitored within the San Joaquin River during the 2013 and 2014 field seasons.

Candidate Model	K	AIC_c	ΔAIC_c	W_i
Intercept + Reach + Mean Temp + Year	4	11482	0	1
Intercept + Reach + Mean Temp	3	13691.2	2209.2	0
Intercept + Mean Temp	2	13847	2365	0
Intercept + Reach + 10 Day Max Temp	3	16573.2	5091.2	0
Intercept + Reach	2	19417	7935	0
Intercept + 10 Day Max Temp	2	19663	8181	0
Intercept + Velocity + Substrate	3	21520.2	10038.2	0
Intercept + Substrate	2	23145	11663	0
Intercept + Velocity	2	23766	12284	0
Intercept + Flow	2	24320	12838	0
Intercept (Null Hypothesis)	1	24322.3	12840.3	0

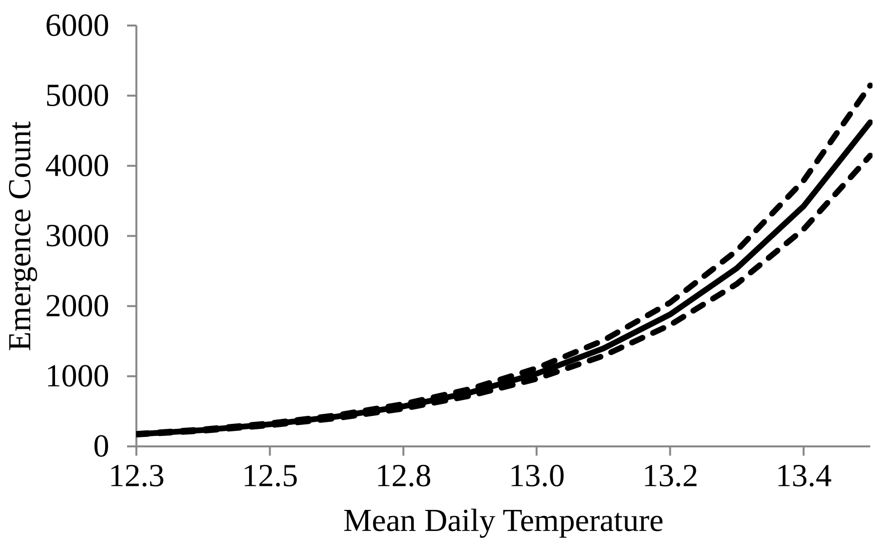

Figure 12. The mean relation between fry emergence count and mean daily temperature within the Highway 41 Reach in the San Joaquin River during the 2013 and 2014 field seasons. The broken lines represent the upper and lower 95% confidence limits. Estimated mean fry emergence count (95% confidence limits; dashed lines) within the Highway 41 Reach related to mean daily water temperatures observed among monitored redds within the San Joaquin River during the 2013 and 2014 field seasons. These estimates were obtained using the best fit candidate model.

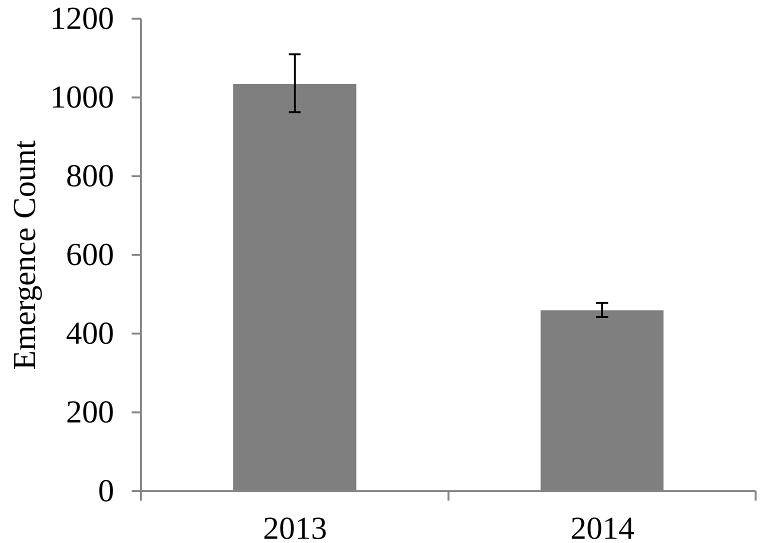

Figure 13. Estimated mean fry emergence count (95% confidence limits) among field seasons within the San Joaquin River. These estimates were obtained using the best fit candidate model and assumed average environmental conditions (mean daily temperature = 12.96°C) and the reach baseline (Highway 41 Reach).

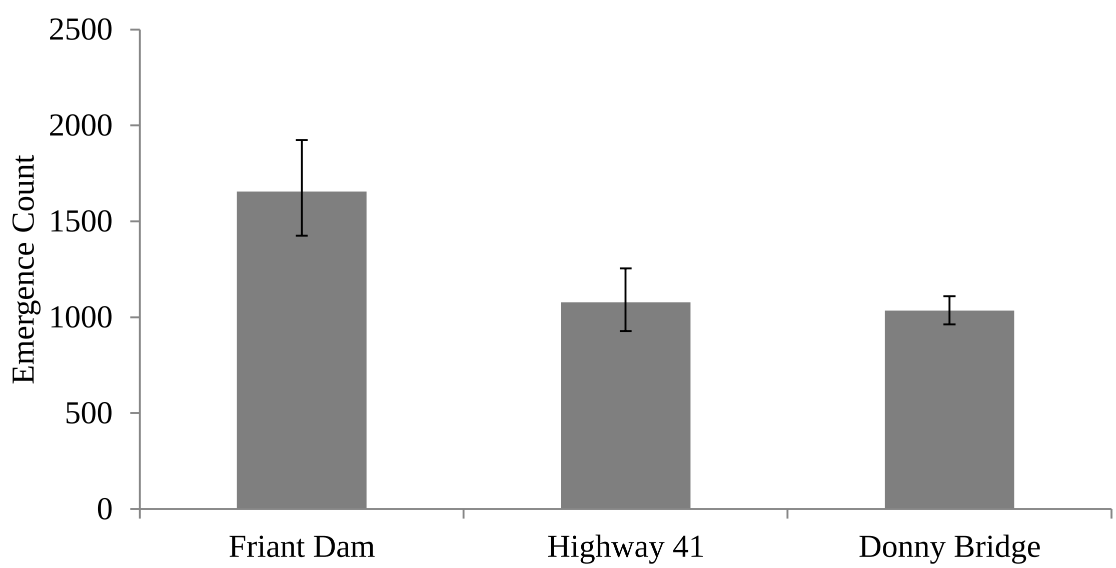

Figure 14. Estimated mean fry emergence count (95% confidence limits) among sample reaches within the San Joaquin River during the 2013 and 2014 field seasons. These estimates were obtained using the best fit candidate model and assumed average environmental conditions (mean daily temperature = 12.96°C).

1.4 Conclusions and Recommendations

Our study revealed that the spawning success of translocated Chinook Salmon in Reach 1 of the Restoration Area during 2013 and 2014 was below the SJRRP 85% target and that the number of fry produced from their redds was below the SJRRP target of a minimum of 50% survival or 2,100 fry per redd. As a result, this study provides evidence that the quality of extant salmon spawning habitat within the Restoration Area may be inadequate for achieving the Chinook Salmon reproduction targets during at least critical (dry) water years. We surmise that the spawning success of translocated Chinook Salmon may have been negatively affected by the condition of translocated salmon, high river temperatures, straying outside of our sampling extent, and/or poaching. Whereas the emergence of fry from redds made by translocated females was likely negatively affected by the condition of translocated salmon coupled with suboptimal environmental conditions that varied among reaches (e.g., quality of spawning gravel) and years (e.g., river flow). We recommend that the SJRRP continue collecting spawning success and representative emergence count data annually to determine to what extent spawning habitat restoration may be needed to achieve the SJRRP salmon population targets (SJRRP 2010). We recommend that this study be continued annually to (1) determine spawning success during different flow and temperature regimes (i.e., among different water year types) and (2) assess the relative importance of environmental variables (e.g., substrate composition) influencing fry emergence. To reduce the uncertainty of our results, we recommend that the SJRRP also tests the validity of our monitoring or analytical assumptions including emergence trap closure and influence on emerging fry or their habitat, and effects of acoustic tag placement on gravid females.

The information generated from this study provides valuable information that can be used to assess the overall suitability of extant salmon spawning habitat occurring in Reach 1 and thereby determine if spawning habitat restoration is needed to achieve fall-run and spring-run Chinook Salmon reintroduction targets. The SJRRP’s Spawning and Incubation Small Interdisciplinary Group (SIG) has recently developed a framework to determine if/how much spawning habitat requires restoration or augmentation by, in part, relating egg survival-to-emergence rates to physical habitat variables/conditions known to influence the salmon egg to fry life-stages (Figure 2). We recommend that our approach and empirical findings be considered by the SIG when assessing the suitability of existing salmon spawning habitat within the Restoration Area.

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